Patriotic Values for Public Goods: Transnational Trade-Offs for Biodiversity and Ecosystem Services?

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The natural environment is central to human well-being through its role in ecosystem service (ES) provision. Managing ES often requires coordination across international borders. Although this may deliver greater conservation gains than countries acting alone, we do not know whether the public supports such an international approach. Using the same questionnaire in three countries, we quantified public preferences for ES in home countries and across international borders. In all three countries, the people were generally willing to pay for ES. However, our results show that there is a limit to the extent that environmental goods can be considered global. ES with a use element (habitat conservation, landscape preservation) attracted a patriotic premium, such that the people were willing to pay significantly more for locally delivered services. Supranational management of ES needs to be balanced against the preferences that people have for services delivered in their home countries.

Keywords: biodiversity conservation, choice experiment, ecosystem services, nonmarket valuation, seminatural grassland, stated preference

The natural environment is central to human well-being through its role in ecosystem service provision (Sachs et al. 2009). There is, therefore, considerable interest in how best to manage the natural world to enhance the delivery of a wide range of services (e.g., Kumar 2010, UKNEA 2011). However, the effective preservation and enhancement of biodiversity and ecosystem services can require intervention across varied socioeconomic and political borders, not least because ecosystems, the biodiversity they contain, and the services they deliver are often shared among such contexts. For example, long-distance migratory species can be responsible for functional links across distant regions (Bauer and Hoye 2014) and, therefore, require novel approaches to their management (e.g., Semmens et al. 2011), which can include transnational organizations. In sub-Saharan Africa, for example, highly mobile migrant pests move frequently across national borders (Dallimer et al. 2003, Cheke and Tratalos 2007). Multinational agencies coordinate management at a regional level to minimize the ecosystem disservices, in the form of crop yield losses, caused by such pests. Elsewhere, supranational bodies, such as the European Union, frame policies and legislation for species and habitat management that operate across many different nations (European Commission 1979, 1992, 2000). Finally, many water catchments are transnational (Lopez-Hoffman et al. 2010) and are managed as such.

Despite the widespread existence of trans- and supranational bodies in ecosystem and biodiversity management, we know little about the extent of public support for initiatives that operate at international scales. This is important because, with limited resources available for biodiversity conservation and ecosystem management, we require an understanding of people's preferences for different aspects of the natural world as one means to prioritize actions for a number of reasons: People have opinions about where to invest in conservation (Jacobsen and Thorsen 2010); conservation is frequently funded by governments who may wish to respond to the values expressed by the public; and interventions are more likely to succeed if they align with public preferences. This raises questions as to the extent to which biodiversity and environmental goods and services should be delivered locally, as well as globally. Some services, such as recreation, landscape appreciation, or wild species diversity, may have a greater value to nearby populations, who are able

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to experience them and therefore benefit from their use—as well as nonuse—values (Atkinson et al. 2012). Others, such as carbon sequestered and storage through vegetation restoration, although they are often quantified at a local scale, deliver their benefits globally (Bulte et al. 2002).

Here, we quantify the values that the public places on biodiversity and ecosystem services delivered across international boundaries, as opposed to within their country of residence. We base our study in the European Union, where many policies pertaining to biodiversity conservation and ecosystem service management (e.g., birds, habitat directives, common agricultural policy and its agrienvironment elements, commitments to reduce carbon emissions) are formulated at a supranational level. Although the available evidence suggests that this approach can be relatively effective at the continental scale at protecting, for example, avian populations (Donald et al. 2007), there is little understanding of the extent to which the general public in Europe supports allocating international funds for ecosystem service management as opposed to a more local approach.

Methods

A commonly used approach to assess public preferences for the natural world is to assign monetary values to changes in ecosystems and the services they supply. Although it is sometimes controversial among conservation biologists, monetary valuation facilitates making a direct comparison with other costs and benefits in decisionmaking processes, and its use has become widespread (Hanley and Barbier 2009, Kumar 2010). Here, we used the stated preference nonmarket valuation technique of the choice experiment (CE) to ask two questions: Do people value ecosystem services and biodiversity across international boundaries, and, if so, how do those values vary according to the scale at which the goods, themselves, deliver benefits? To do this, we chose a suite of services that vary in their scale of delivery from global (enhanced carbon capture for climate change mitigation) through both global and local (biodiversity conservation) to mainly local (the preservation of landscapes that are culturally and aesthetically appreciated; see the "Survey design" section below). We hypothesize that there will be a preference for ecosystem services to be delivered locally, as opposed to across international borders, and that this preference will be weaker for more-global public goods.

Choice experiments draw on theories of economic value (Lancaster 1966) and the application of random utility theory to choice (McFadden 1974). The methodology is based on probabilistic choice, in which individuals select a single alternative from a set of available alternatives that are presented to them. The attributes of the chosen alternative (here of various ecosystem services) are assumed to maximize the individual's utility (supplemental appendix S1). Choice experiments involve presenting participants with a number of choice sets that consist of two or more alternatives, each described by various levels of a set of attributes and a monetary cost that would finance the changes in the attribute levels described in an alternative. This allows the level of willingness to pay (WTP) to be calculated using estimated parameters of the choice probability function for the different alternatives. The WTP for a marginal improvement in an attribute can then be calculated as the ratio between the parameter of that attribute and the parameter of the price attribute (see the supplemental material for analytical details). Choice experiments are commonly used to valuate changes in ecosystem services and biodiversity (Christie et al. 2006, Jacobsen and Thorsen 2010, Morse-Jones et al. 2012, Dallimer et al. 2014) and offer a wide range of information on the trade-offs among the benefits provided by the different alternatives (Adamowicz et al. 1997, 1998).

Survey design

The focus of the CE was to valuate changes in ecosystem services across international borders. We used seminatural grasslands in northern Europe, a study system for which such an analysis is particularly pertinent, not least because environmental policy delivered across the member states of the European Union has a long-standing international component (e.g., the Birds and Habitats Directives and the Natura 2000 network of protected areas; European Commission 1979, 1992, 2000). Seminatural grasslands have historically been subject to huge losses in extent and quality (Veen et al. 2009), and they are important for cultural and aesthetic reasons (e.g., Sand-Jensen 2007), as well as being a key habitat for biodiversity conservation in Europe. This was acknowledged by Mariann Fischer Boel, the EU Commissioner for Agriculture and Rural Development, in 2009, "grasslands… represent a key element in Europe's rich diversity of landscapes, and the public appreciate[s] the beauty of Europe's meadows" (Veen et al. 2009). Indeed, many grassland systems are included in the continent's register of high nature value farmland, which recognizes the central place that traditional farming techniques play in maintaining culturally important and biodiverse landscapes (e.g., Knowles 2011). Despite this, and even though they deliver a wide range of ecosystem services (European Commission 2008), grasslands are rarely the subject of nonmarket valuation exercises.

We selected attributes for the CE based on services that are delivered by seminatural grasslands, have an international dimension to their management, and are likely to span different scales of beneficiaries. Three such services are the preservation of landscapes that are culturally and aesthetically appreciated, biodiversity conservation, and enhanced carbon capture for climate change mitigation.

The European Union promotes the preservation of landscapes through the European Landscape Convention (Council of Europe 2000). Regions with a high coverage of seminatural grasslands often retain features associated with culturally important and aesthetically attractive landscapes, such as traditional buildings, boundaries, and field sizes (Sand-Jensen 2007, Veen et al. 2009, Knowles 2011). Traditional landscapes tend to have strong cultural links to the region in which they are found (Jacobsen and Thorsen 2010), and their enjoyment is therefore largely a use value. We would expect beneficiaries to be mainly restricted to the country in which a particular region is located.

The conservation of biodiversity and habitats within the European Union is governed via instruments such as the Habitats Directive (European Commission 1992), which all member states are expected to implement. Biodiversity is considered central to supporting all ecosystem services (Balvanera et al. 2006). However, there is an ongoing debate as to whether biodiversity *per se* can be considered a service in and of itself (Mace et al. 2012), although the protection of biodiversity clearly has value to people (e.g., Christie et al. 2006, Morse-Jones et al. 2012, Dallimer et al. 2014). For example, the UK National Ecosystem Assessment includes wildlife diversity, both as an intermediate service and as a final provisioning and cultural service (UKNEA 2011). We include it as a final service because of its associated use and nonuse values for EU citizens (e.g., UKNEA 2011, Bateman et al. 2013). The benefits of the service could therefore be experienced potentially both locally and globally.

The European Union has committed its member states to reducing carbon emissions by 20% below 1990 levels by 2020 (EEA 2010). Enhancing storage and uptake within vegetation and soils is one potential pathway through which part of these targets could be met. Seminatural grasslands can be managed by manipulating fertilizer application and grazing levels and by promoting the presence of certain forbs to increase carbon uptake and storage in some situations (De Deyn et al. 2011). The benefits delivered by this service (in terms of climate amelioration) would be experienced globally.

We elected to use an increase in the area (measured in hectares) under management for biodiversity as an attribute rather than an increase in species richness or the abundance of key species. This was to ensure that our estimates of WTP would not be affected by preferences for certain taxa (e.g., Jacobsen et al. 2008). The landscape preservation attribute was also hectare based, making it directly comparable to the biodiversity conservation attribute. However, the units for the carbon capture attribute were tonnes of carbon per hectare per year. Although these units are perhaps more abstract than a third hectare-based attribute, the direct benefit to people from the carbon attribute is the amount in tonnes of carbon captured rather than the number of hectares over which the carbon is distributed. We, therefore, used the component that carries the utility directly, even though this may restrict direct comparisons of value between the different services.

Our study system was centered on northern Europe. Within this region, we selected regions that were comparable in terms of topography, area, habitat type, and the number and extent of designations under the EU Habitats Directive (supplemental appendix S1). We also wished to cover a range of international cultural differences found in this region and, therefore, included a Western European nation (Denmark), a former communist country (Poland), and a former constituent of the Soviet Union (Estonia; figure 1). By choosing sites that were similar, we attempted to ensure that the CE quantified transnational effects on the values that people ascribe to the sites, rather than, for example, habitat preferences, marginal effects related to how large our example regions were, or preformed preferences for certain locations or species (Jacobsen et al. 2008, Bateman 2009, Jacobsen and Thorsen 2010).

To estimate measures of economic benefit from changes in the environmental attributes listed above, a cost attribute was included in the design, specified as an increase to the householder's annual taxation bill needed to finance the management measures. The choices would then show how much people are willing to trade off improvements in an environmental attribute for a decrease in their income. The levels of the cost attribute were determined on the basis of previous studies (Bartczak et al. 2008, Jacobsen and Thorsen 2010) and were adjusted following focus groups and pilot tests. Each nationality was presented with costs in their local currency, with the amounts' purchasing power parity calibrated to be equivalent.

An optimal design for the CE was generated, and we included Bayesian priors from a pilot exercise to improve design efficiency (Ferrini and Scarpa 2007, Scarpa and Rose 2009). This resulted in a CE consisting of 12 choice cards, divided into two blocks. Each respondent therefore faced six choice sets, which asked them to choose between four alternatives (for an example, see supplemental appendix S2). These were the three *policy-on* options, which included different combinations of the attributes (carbon capture, habitat conservation, landscape preservation, region, and the annual tax cost), and a no-cost status quo alternative, in which no changes would take place across all regions. The policy-on options included the baseline of no change and two levels of change in carbon capture, habitat conservation, and landscape preservation and six levels of cost (table 1).

The questionnaire was initially developed in English and was translated by native speakers into the relevant local languages. We used focus groups and a pilot exercise to help finalize the questionnaire in two different ways. First, feedback from the participants ensured that the translations were understandable to the general population and used appropriate wordings that were relevant to national situations. Final versions of the questionnaire were therefore produced only in Danish, Polish, Estonian, and Russian (to account for the Russian-speaking population in Estonia) and are available from the authors. Second, the focus groups and pilot exercises allowed us to test the structure and meaning of the CE and its associated attitudinal and sociodemographic questions.

Commercial polling companies were used to deliver the survey to an online panel of respondents in winter 2012. Around 3200 individuals were invited to take part in the survey in each country. Data collection was finalized when at least 850 respondents (representative of the national population according to age, gender, education, and employment) had completed the questionnaire. Initially, we were supplied with over 1200 responses from Poland, but we wished to

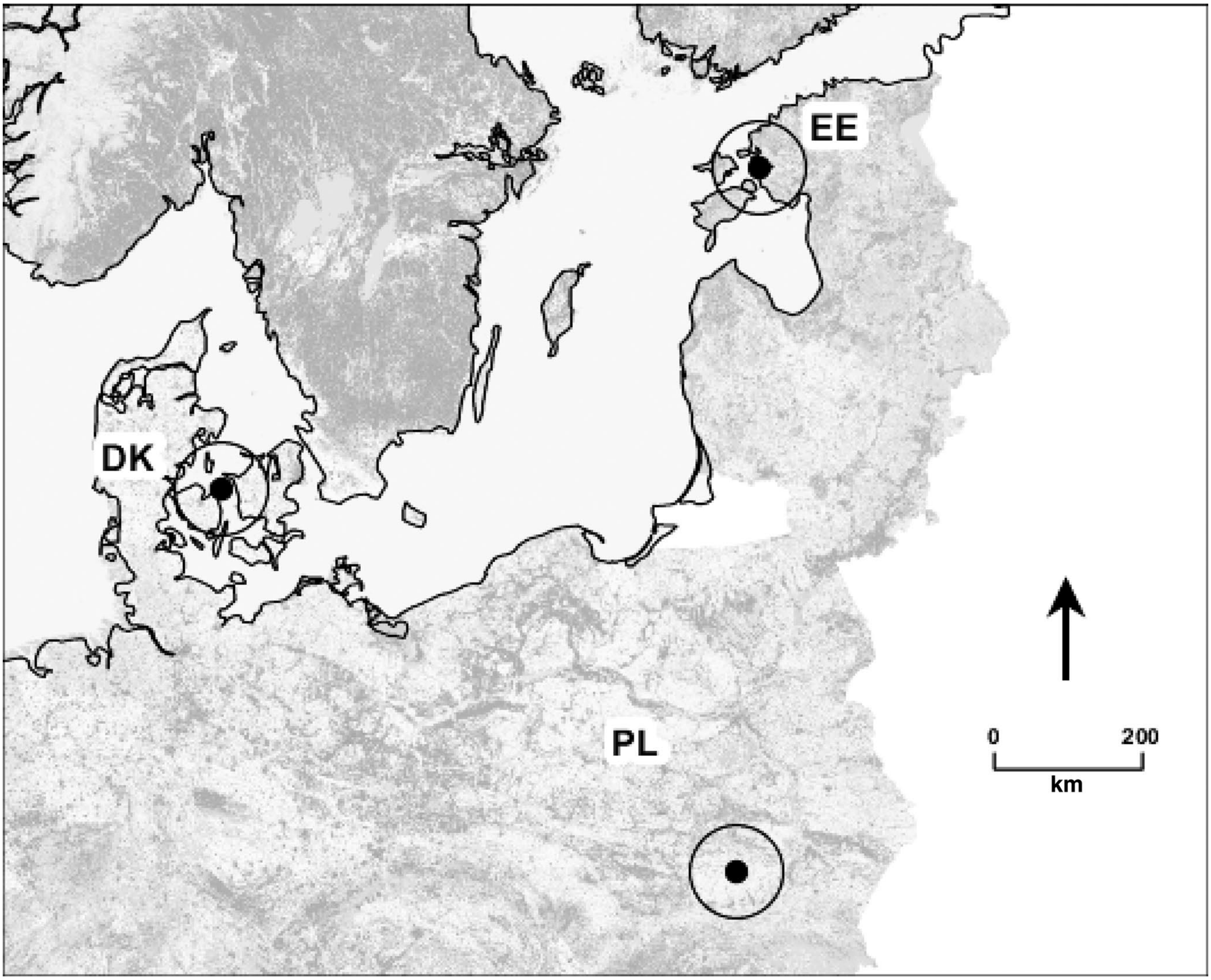

Figure 1. Northern Europe, showing the location of the study regions within Denmark (DK), Poland (PL), and Estonia (EE). For site descriptions as presented to the respondents, see supplemental appendix S1. Abbreviation: km, kilometers.

have an equivalent number of respondents in each country, so a random sample was selected from these to bring the sample size in line with those from Denmark and Estonia. Of the completed responses, we removed 22 (0.8%) that were completed in less than 5 minutes (insufficient time to read through the survey) and 25 (1.0%) for which answers to the full set of choice cards were not recorded. The status quo option was chosen for all choice cards by 138 (5.4%) respondents who also gave a motivation for this pattern of answers that was consistent with protesting against the questionnaire itself or the payment vehicle used (supplemental appendix S3). Although the proportion of protesters was small, standard practice assumes that they did not reveal their true preferences and that they should be excluded from further analyses (Meyerhoff and Liebe 2008, Jacobsen and Thorsen 2010). The remaining data from all countries were merged and analyzed together, resulting in a final sample size of 2367 (approximately 800 respondents per country, answering 14202 choice cards). The analyses were conducted in NLOGIT (Econometric Software, Plainview, New York) using a mixed logit specification with an error component model (Scarpa Riccardo et al. 2005, Greene and Hensher 2007). Parameter estimates from the simpler conditional logit model were of the same sign and magnitude as those from the mixed logit, so we report only the results from the more complex model. We included a correction for scale difference (Hensher et al. 1999) among nationalities. Details of the analytical approach and theoretical background are given in supplemental appendix S1.

Results

The respondents of all nationalities expressed a positive and significant WTP for enhanced ecosystem services (table 2). Irrespective of where the services were to be delivered, the respondents stated a mean (*M*) WTP for habitat conservation of €0.038 (standard error [SE] = 0.004) and for

Table 1. Attributes and levels presented in the choice experiment to determine willingness to pay for ecosystem services delivered across international borders in the European Union.

Attribute	Levels	Status quo
Carbon capture	2 or 3 tonnes carbon captured per ha per year	1 ton carbon captured per ha per year
Habitat conservation	An extra 500 or 1000 hectares of seminatural grassland managed for wildlife and habitat conservation	No change
Landscape preservation	An extra 500 or 1000 hectares of traditional landscape preserved	No change
Region	Changes only take place in Denmark, Poland, or Estonia	No change in any region
Price	0, 100, 200, 400, 800, 1200 Dkr. (Denmark); 0, 25, 50, 100, 225, 350 zł (Poland); 0, 5, 10, 25, 55, 85 € (Estonia)	0 Dkr, zł, or €

Table 2. Parameter and willingness to pay (WTP) estimates for a random parameter error component logit model for the main effects model, based on 14,202 observations from 2367 respondents (χ^2 = 9102.99, pseudo-R^2 = .231, log-likelihood = –15137.15).

	Parameter		Standard deviation		WTP	
Variable	Value	Standard error (SE)	Value	SE	Value (in €)	SE
ASC	0.869***	0.118			0.078***	0.011
Estonia	–0.246***	0.062	1.783***	0.063	–0.022***	0.006
Poland	–0.082	0.047	1.393***	0.054	–0.007*	0.004
Habitat conservation	0.427***	0.049	0.855***	0.073	0.038***	0.004
Landscape preservation	0.313***	0.045	0.497***	0.089	0.028***	0.004
Carbon capture	0.210***	0.022	0.182***	0.053	0.019***	0.002
Price	–1.507***	0.029				

Note: The simulations were based on 1000 Halton draws. The alternative specific constant (ASC) is confounded with the benchmark region of Denmark, and the estimates for Estonia and Poland are additional to it. WTP is reported in € per household per year for management interventions to take place over 1 hectare. For carbon, WTP is per ton of carbon captured on that hectare. *p ≤ .1. *p ≤ .05. *p ≤ .01.

landscape preservation of €0.028 (SE = 0.004) per household per year for the management of one additional hectare. The mean WTP for carbon capture was €0.019 (SE = 0.002) per household per year for an extra tonnes of carbon per hectare.

There were significant preferences for where management actions should take place. The respondents from the complete sample expressed the highest preference for actions in Denmark (as contained within the alternative specific constant [ASC]), with a mean WTP value of €0.078 (SE = 0.010). The variable ASC measures the WTP for taking any form of action (relative to the status quo) irrespective of the outcomes of those actions. Given that country variables are 0/1 dummies, in order for us to carry out the estimation and not overspecify our models, we did not include one country—in this case, Denmark. The WTP amounts for Denmark were therefore confounded with the ASC. The WTP measures for Poland and Estonia are relative to the ASC. Therefore, across all respondents, the WTP for management actions in Poland was €0.007 (SE = 0.004) lower than that in Denmark, and ecosystem services delivered in Estonia were on average significantly less valued across the respondents from the three countries, being €0.022 (SE = 0.006) lower than that in Denmark. The overall utility for actions in Estonia was still positive and significantly different from zero. This pattern reflects that all of the respondents were more likely to choose alternatives based in their own country and that the Polish and Danish respondents chose alternatives in Denmark and Poland, respectively, more often than they chose provision in Estonia. Similarly, Estonians were largely indifferent in their choices between Denmark and Poland (table 3).

Although prices were purchasing power parity corrected, we would still expect there to be significant differences among nationalities with respect to the marginal utility of income. We accounted for this by including two nationality × price interaction variables in the models. As was previously noted, because the country variables are 0/1 dummies, we could only include two of them in the model. Therefore, the parameter estimate for price refers to that of the Danish respondents, and the interaction terms for Poland and Estonia quantify the additional contribution with respect to that price parameter (e.g., for Poland, −1.361 − 0.309 = −1.670). The marginal utility of income was therefore significantly higher for the Polish and Estonian respondents than that for the Danes (table 4; Estonian × price and Polish × price interactions). Because WTP is calculated by dividing the parameter estimate for the environmental attributes by that of price, the precise WTP estimates vary by a fixed ratio among nationalities. For simplicity in the text, we report WTP based on Danish price sensitivity (the WTP values for the Danish respondents in table 4).

Table 3. Frequency with which alternatives involving the named regions were selected by respondents of each nationality.

		Region		
Nationality	Status quo	Denmark	Estonia	Poland
All respondents	0.31	0.22	0.19	0.28
Danish	0.36	0.39	0.10	0.15
Estonian	0.41	0.12	0.36	0.11
Polish	0.21	0.18	0.14	0.47

Note: Status quo indicates that the no change option was selected.

We wished to separate out the effects of nationality and region to examine the more general issue of how much extra people were willing to pay to have a service delivered in their own country, rather than the exact same service provided elsewhere. We did this by including a variable for the respondents' own country (which took the value 1 when management actions took place in the respondent's country of residence and 0 otherwise), which interacted with the environmental attributes. In addition, we included interactions between this variable and the region of provision, which were intended to capture latent and unobserved effects of the respondent's nationality on their preferences. The general pattern remained (table 4, figure 2), with the mean WTP values for habitat conservation, landscape preservation, and carbon capture being €0.034 (SE = 0.007), €0.018 (SE = 0.006), and €0.011 (SE = 0.003), respectively.

The own-country region preferences were all significantly different from zero and positive (table 4), which indicates that the respondents were willing to pay more for any actions to take place in the country in which they resided (figure 2). This was especially marked for the Estonians, who were willing to pay an additional €0.114 (SE = 0.015) for actions in Estonia. In contrast, the Danes expressed the lowest additional valuation for actions to take place in their own country, with a mean WTP of €0.033 (SE = 0.013).

Across all three countries, the WTP values for habitat conservation and landscape preservation within the respondents' own countries more than doubled the WTP estimate for the same actions undertaken elsewhere. For example, the mean WTP for habitat conservation was €0.034 (SE = 0.007), whereas the additional WTP for habitat conservation in a respondent's home country (as captured by the own country × habitat conservation interaction) was €0.047 (SE = 0.011), giving a total WTP for habitat conservation of €0.081. The own-country patriotic premium was largest for landscape preservation. The premium for carbon capture delivery in a

Table 4. Parameter and willingness to pay (WTP) estimates for a random parameter error component logit model for the own-country model, based on 14,202 observations from 2367 respondents (χ^2 = 11066.51, pseudo-R^2 = .281, log-likelihood = –14154.9).

	Parameter		Standard deviations		Danish respondents		Estonian respondents		Polish respondents	
Variable	Value	Standard error (SE)	Value	SE	WTP (in €)	SE	WTP (in €)		WTP (in €)	SE
ASC	0.588***	0.128			0.058***	0.012	0.054***	0.011	0.048***	0.010
Estonia[a]	–0.370***	0.073	1.024***	0.061	–0.037***	0.007	–0.034***	0.007	–0.030***	0.006
Poland[a]	–0.122	0.074	0.842***	0.051	–0.012***	0.007	–0.011***	0.007	–0.010***	0.006
Habitat conservation	0.342***	0.071	0.734***	0.071	0.034***	0.007	0.031***	0.007	0.028***	0.006
Landscape preservation	0.183***	0.065	0.590***	0.074	0.018***	0.006	0.017***	0.006	0.015***	0.005
Carbon capture	0.111***	0.031	0.126*	0.061	0.011***	0.003	0.010***	0.003	0.009***	0.003
Own country × habitat conservation	0.478***	0.111	–	–	0.047***	0.011	0.044***	0.010	0.039***	0.009
Own country × landscape preservation	0.412***	0.102	–	–	0.041***	0.010	0.038***	0.009	0.033***	0.009
Own country × carbon capture	0.161***	0.047	–	–	0.016***	0.005	0.015***	0.004	0.013***	0.004
Own country × Denmark	0.332**	0.134	–	–	0.033**	0.013	–	–	–	–
Own country × Estonia	1.245***	0.162	–	–	–	–	0.114***	0.015	–	
Own country × Poland	0.970***	0.105	–	–	–	–	–	–	0.079***	0.009
Estonian × price[b]	–0.115	0.063	–	–	–	–	–	–	–	–
Polish × price[b]	–0.309***	0.077	–	–	–	–	–	–	–	–
Price	–1.361***	0.047	–	–	–	–	–	–	–	–

Note: WTP estimates are presented for each nationality, calculated from the appropriate price parameter. The WTP for each attribute and country was calculated using the preference parameter for the attribute divided by the country's marginal utility of income—for example, –1.361 for Denmark and –1.670 (–1.361 – 0.309) for Poland. The WTP values are given in € using the conversion rate of 7.4 Dkr/€. Abbreviation: ASC, alternative specific constant. [a]As compared with management action in Denmark. [b]Additional to the price. *p ≤ .1. *p ≤ .05. *p ≤ .01.

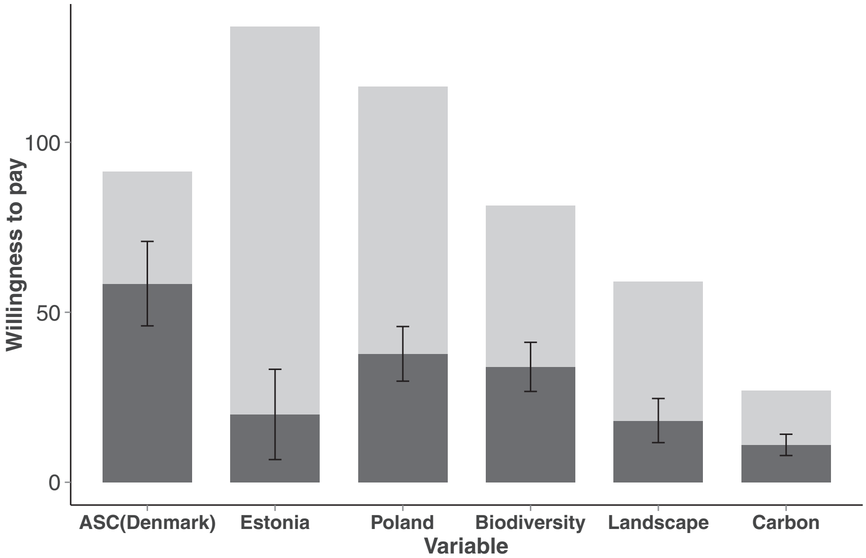

Figure 2. Willingness to pay (WTP; in euros per household per year) for management action over 1000 hectares for the own-country interactions model (table 4). The light grey bars indicate the amount that the participants were willing to pay for actions carried out in their country of residence in addition to the WTP estimate (in dark grey) for actions not taking place in their country of residence. The error bars represent standard errors.

respondent's own country was smallest, although it was still of significant size (table 4, figure 2).

Thus far, our results support our two main hypotheses—namely, that there should be a preference for ecosystem services to be delivered locally, as opposed to across international borders, and that this preference should be weaker for more-global public goods. However, there are other potential explanations for the patterns so far described. For instance, the preference for services delivered in a respondent's country of residence could be driven by regular outdoor recreationalists' being willing to pay higher amounts for locally delivered services for which they gain use value. We accounted for this by including a variable for frequent (more than one visit per month) recreational visitors to the countryside. Finally, although we used purchasing power parity to match tax amounts presented to the respondents from different countries, we would expect the respondents on relatively high incomes to exhibit a different sensitivity to price than those on low incomes. We controlled for this by including an interaction between price and a high income respondents (i.e., those whose household incomes were in the upper income brackets for their country of residence; supplementary appendix S4).

The respondents reporting household incomes in the higher brackets for their country and regular recreational users were less sensitive to price (table 5; high income × price interaction, M = €0.171, SE = 0.051; user × price interaction, M = €0.181, SE = 0.051). Although regular users had a generally higher WTP (the user × price term), they were not willing to pay a greater amount for any specific environmental attributes (none of the parameter estimates for the user × habitat conservation, user × landscape preservation, user × carbon capture interactions were significantly different from zero). There was no impact on the magnitude or relative ranking of the preferences for services to be delivered in the respondents' own countries (table 5).

Discussion

Across three European countries, we found a significant WTP for enhancements to ecosystem services provided by seminatural grasslands, regardless of the location of delivery (table 1). Nevertheless, the respondents were willing to pay significantly greater amounts for services located in their country of residence (tables 3 and 4). The magnitude of this extra payment was linked to the extent to which the good could be considered local or global. The additional WTP for services with characteristics of a local public good (in our study, habitat conservation and landscape preservation) to be delivered within the respondents' countries of residence was much higher than that for the global public good of carbon capture.

Given that local goods are assumed to have a high use value, perhaps surprisingly, we did not find that regular recreational users of the countryside were willing to pay more for locally delivered services (although they did have a higher WTP values across all services and locations in general). Nonuse values can be experienced by people without engaging in specific activities and behaviors. We may, for example, all derive utility from knowing that endangered species are protected, even though we may never see them (e.g., Morse-Jones 2012). Such values require no measurable action for us to experience and are likely to be global in nature, because they are nonrival and noone can be excluded from receiving their benefits. In contrast, use values are accrued through active use, including activities such as wildlife watching and enjoying aesthetically pleasing landscapes. Because use values imply a cost for the user, in terms of money, transport, and time, people are likely to care about where and how they can be enjoyed. Therefore, the values of environmental public goods with large use components are likely to be less global in nature.

By simultaneously considering both respondents from and ecosystem service delivery within several countries, we demonstrated a strong preference for local delivery and for the value that people can attach to services provided outside their home country. Cultural heritage, shared values, and experiences can affect values for public goods (Ready and Navrud 2006, Jacobsen and Thorsen 2010). Here, respondents

Table 5. Parameter and willingness to pay (WTP) estimates for a random parameter error component logit model for the frequent user model, based on 12,498 observations from 2083 respondents (χ^2 = 9743.25, pseudo-R^2 = .281, log-likelihood = –12454.3).

	Parameter		Standard deviations	
Variable	**Value**	**Standard error (SE)**	**Value**	**SE**
ASC	0.637***	0.135		
Estonia[a]	–0.384***	0.078	1.028***	0.065
Poland[a]	0.126	0.081	0.081***	0.054
Habitat conservation	0.331***	0.085	0.712***	0.077
Landscape preservation	0.096	0.081	0.575***	0.081
Carbon capture	0.146***	0.039	0.141*	0.063
Own country × habitat conservation	0.518***	0.120		
Own country × landscape preservation	0.415***	0.111		
Own country × carbon capture	0.145***	0.051		
Own country × Denmark	0.376***	0.145		
Own country × Estonia	1.293***	0.175		
Own country × Poland	0.978***	0.113		
Estonian × price[b]	–0.136*	0.067		
Polish × price[b]	–0.326***	0.083		
User × habitat conservation[b]	0.020	0.086		
User × landscape preservation[b]	0.129	0.080		
User × carbon capture[b]	–0.036	0.040		
User × price[b]	0.181***	0.051		
High income × price[b]	0.171***	0.051		
Price	–1.528***	0.065		

Note: We do not present WTP as price parameters that differ significantly across many different subgroups. ASC takes the value 1 for the alternative and is, therefore, confounded with the reference level of management action in Denmark. Abbreviation: ASC, alternative specific constant. [a]As compared with management action in Denmark. [b]Additional to the price. *$p \le .1$. *$p \le .05$. *$p \le .01$.

in Denmark, Poland, and Estonia were willing to pay significantly different amounts for management to enhance ecosystem services, which suggests that nationality and international borders were important determinants of value. Nevertheless, political boundaries are not the same as market boundaries when assessing WTP for environmental enhancements (Loomis and White 1996). For example, residents in developed countries are willing to pay for the conservation of species in the developing world (Morse-Jones et al. 2012), and the optimal coverage by rainforest in Costa Rica is markedly higher when global (as well as local) beneficiaries are included in the calculations (Bulte et al. 2002). Similarly, nationality is not always a strong determinant of value (Jin et al. 2010).

Since their popularization (MA 2005), ecosystem services have gained considerable traction among researchers and policymakers keen to incorporate values for the natural world in decisionmaking processes (UKNEA 2011, Bateman et al. 2013). Although biodiversity has a role in underpinning many services (Atkinson et al. 2012, Mace et al. 2012), there is a danger that biodiversity conservation *per se* will be overlooked in the face of more obviously beneficial and quantifiable services, such as climate mitigation. However, biodiversity plays an important role in delivering cultural services (Mace et al. 2012) and is highly valued by the general public (Christie et al. 2006, Morse-Jones et al. 2012, Dallimer et al. 2014). Across the three countries in our study, when they were faced with a choice between management for biodiversity conservation and two other services, the respondents consistently placed higher values on biodiversity, which indicates that it should retain a prominent role in environmental management and policy.

We acknowledge competing explanations for the pattern documented here, not least because many other variables may be entirely confounded with region and nationality and could therefore weaken the patterns that we have quantified. For example, it is possible that the size of the chosen regions was an important factor in the respondents' WTP values for management actions focused on particular locations. We addressed this by ensuring that the study regions were closely matched in terms of their existing areas of seminatural grassland. However, there remained substantial differences in the number of species considered to be under threat of extinction among the study sites (47, 54, and 22 for Estonia, Poland and Denmark, respectively; see the supplemental material). The fact that the Danes expressed the lowest additional WTP value for habitat conservation actions to take place in their own country could plausibly be driven by the perception that actions in Denmark would contribute least to biodiversity protection across the three countries. Similarly, although the respondents were not presented with the information, the relative rarity of the habitat and landscapes in each country may have played a role. For example, if a habitat is thought to be rare in a certain country, the marginal benefits of increasing coverage may be greater than that in a country in which the habitat is perceived to be common. In our study, this would translate to the respondents' demonstrating a preference for investment in habitat conservation in Denmark, where seminatural grasslands are relatively scarce compared with either Poland or Estonia. A further plausible hypothesis might be that people factor into their preference the relative costs across our three study countries. In this case, Denmark, where prices and incomes are highest, would be perceived to be the most costly country in which to undertake management actions, and therefore, the respondents may feel that their WTP would need to be greater to deliver the same environmental changes. In both cases within our CE, this would result in higher WTP estimates for actions carried out in Denmark or via a reduced preference for the respondents' own countries among the Estonian and Polish respondents. Although we did not see the latter, the WTP estimate for any action to take place in Denmark (as captured by the ASC) was higher than those for Poland or Estonia (table 1).

Finally, the preferences for public goods delivered across international borders may be influenced by the varying levels of trust that exist both within and between people and institutions of different nationalities (e.g., Zak and Knack 2001). For example, the Estonians may have believed that their own country, with its associated laws, compliance, and governance structures, is more likely to deliver enhanced ecosystem services than is either Denmark or Poland (and vice versa). Alternatively, they may feel more in control of implementation if management is carried out locally (Hanley et al. 2003).

Conclusions

The current prioritization of conservation efforts tends to incorporate biophysical variables together with information regarding the distribution of the socioeconomic costs of land management (Ando et al. 1998, Bode et al. 2008). Large-scale, often supranational prioritization may well be the most efficient way to deliver maximum conservation gain (Bladt et al. 2009, Kark et al. 2009). However, this takes no account of how benefits from conservation management that accrue to the human population are distributed.

A supranational approach to ecosystem management has some support among the general population. However, the values that people express for ecosystem goods and services delivered internationally need to be balanced against the substantially higher WTP for services that are enhanced in their country of residence. Such a finding has important implications for how environmental management and biodiversity conservation are prioritized. The distinct preferences for locally delivered ecosystem services could imply a lower acceptance of international cooperation on environmental issues, coupled with a greater demand for investments in environmental programs in one's own country. In particular, goods with an obvious use value (e.g., biodiversity, aesthetically pleasing landscapes) cannot be considered as truly global public goods. In our study system, as in many others, this raises issues of trust between countries, because the potential for free riding is high. Ecosystem management could proceed in Poland, financed solely by Polish taxes, but the people in nearby countries would also benefit. In many other cases, services are shared across international boundaries (e.g., carbon sequestration, catchment-level water quality, and migratory species), and cooperative management would be required to maximize their value to the residents of all countries.

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Supplemental material

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References cited

Adamowicz W, Boxall P, Williams M, Louviere J. 1998. Stated preference approaches for measuring passive use values: Choice experiments and contingent valuation. American Journal of Agricultural Economics 80: 64–75.

Adamowicz W, Swait J, Boxall P, Louviere J, Williams M. 1997. Perceptions versus objective measures of environmental quality in combined revealed and stated preference models of environmental valuation. Journal of Environmental Economics and Management 32: 65–84.

Ando A, Camm J, Polasky S, Solow A. 1998. Species distributions, land values, and efficient conservation. Science 279: 2126–2128.

Atkinson G, Bateman IJ, Mourato S. 2012. Recent advances in the valuation of ecosystem services and biodiversity. Oxford Review of Economic Policy 28: 22–47.

Balvanera P, Pfisterer AB, Buchmann N, He JS, Nakashizuka T, Raffaelli D, Schmid B. 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters 9: 1146–1156.

Bartczak A, Lindhjem H, Navrud S, Zandersen M, Zylicz T. 2008. Valuing forest recreation on the national level in a transition economy: The case of Poland. Forest Policy and Economics 10: 467–472.

Bateman IJ. 2009. Bringing the real world into economic analyses of land use value: Incorporating spatial complexity. Land Use Policy 26: S30–S42.

Bateman IJ, et al. 2013. Bringing ecosystem services into economic decision-making: Land use in the United Kingdom. Science 341: 45–50.

Bauer S, Hoye BJ. 2014. Migratory animals couple biodiversity and ecosystem functioning worldwide. Science 344: 54.

Bladt J, Strange N, Abildtrup J, Svenning JC, Skov F. 2009. Conservation efficiency of geopolitical coordination in the EU. Journal for Nature Conservation 17: 72–86.

Bode M, Wilson KA, Brooks TM, Turner WR, Mittermeier RA, McBride MF, Underwood EC, Possingham HP. 2008. Cost-effective global conservation spending is robust to taxonomic group. Proceedings of the National Academy of Sciences 105: 6498–6501.

Bulte E, van Soest DP, van Kooten GC, Schipper RA. 2002. Forest conservation in Costa Rica when nonuse benefits are uncertain but rising. American Journal of Agricultural Economics 84: 150–160.

Cheke RA, Tratalos JA. 2007. Migration, patchiness, and population processes illustrated by two migrant pests. BioScience 57: 145–154.

Christie M, Hanley N, Warren J, Murphy K, Wright R, Hyde T. 2006. Valuing the diversity of biodiversity. Ecological Economics 58: 304–317.

Council of Europe. 2000. The European Landscape Convention. Office for Official Publications of the European Communities.

Dallimer M, Jones PJ, Pemberton JM, Cheke RA. 2003. Lack of genetic and plumage differentiation in the red-billed quelea Quelea quelea across a migratory divide in southern Africa. Molecular Ecology 12: 345–353.

Dallimer M, Tinch D, Hanley N, Irvine KN, Rouquette JR, Warren PH, Maltby L, Gaston KJ, Armsworth PR. 2014. Quantifying preferences for the natural world using monetary and nonmonetary assessments of value. Conservation Biology 28: 404–413.

De Deyn GB, Shiel RS, Ostle NJ, McNamara NP, Oakley S, Young I, Freeman C, Fenner N, Quirk H, Bardgett RD. 2011. Additional carbon sequestration benefits of grassland diversity restoration. Journal of Applied Ecology 48: 600–608.

Donald PF, Sanderson FJ, Burfield IJ, Bierman SM, Gregory RD, Waliczky Z. 2007. International conservation policy delivers benefits for birds in Europe. Science 317: 810–813.

[EEA] European Environment Agency. 2010. Tracking Progress towards Kyoto and 2020 Targets in Europe. EEA.

European Commission. 1979. Council Directive 79/409/EEC of 2 April 1979 on the Conservation of Wild Birds. Office for Official Publications of the European Communities.

———. 1992. Council Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora. Office for Official Publications of the European Communities.

———. 2000. Managing Natura 2000 Sites: The Provisions of Article 6 of the 'Habitats' Directive 92/43/EEC. Office for Official Publications of the European Communities.

———. 2008. LIFE and Europe's Grasslands: Restoring a Forgotten Habitat. Office for Official Publications of the European Communities.

Ferrini S, Scarpa R. 2007. Designs with a priori information for nonmarket valuation with choice experiments: A Monte Carlo study. Journal of Environmental Economics and Management 53: 342–363.

Greene WH, Hensher DA. 2007. Heteroscedastic control for random coefficients and error components in mixed logit. Transportation Research E 43: 610–623.

Hanley N, Barbier E. 2009. Pricing Nature: Cost–Benefit Analysis and Environmental Policy. Edward Elgar.

Hanley N, Schlapfer F, Spurgeon J. 2003. Aggregating the benefits of environmental improvements: Distance-decay functions for use and non-use values. Journal of Environmental Management 68: 297–304.

Hensher D, Louviere J, Swait J. 1999. Combining sources of preference data. Journal of Econometrics 89: 197–221.

Jacobsen JB, Thorsen BJ. 2010. Preferences for site and environmental functions when selecting forthcoming national parks. Ecological Economics 69: 1532–1544.

Jacobsen JB, Boiesen JH, Thorsen BJ, Strange N. 2008. What's in a name? The use of quantitative measures versus 'iconised' species when valuing biodiversity. Environmental and Resource Economics 39: 247–263.

Jin JJ, Indab A, Nabangchang O, Truong DT, Harder D, Subade RF. 2010. Valuing marine turtle conservation: A cross-country study in Asian cities. Ecological Economics 69: 2020–2026.

Kark S, Levin N, Grantham HS, Possingham HP. 2009. Between-country collaboration and consideration of costs increase conservation planning efficiency in the Mediterranean Basin. Proceedings of the National Academy of Sciences 106: 15368–15373.

Knowles B. 2011. Mountain Hay Meadows: Hotspots of Biodiversity and Traditional Culture. Society of Biology.

Kumar P. 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan.

Lancaster KJ. 1966. New approach to consumer theory. Journal of Political Economy 74: 132–157.

Loomis JB, White DS. 1996. Economic benefits of rare and endangered species: Summary and meta-analysis. Ecological Economics 18: 197–206.

Lopez-Hoffman L, Varady RG, Flessa KW, Balvanera P. 2010. Ecosystem services across borders: a framework for transboundary conservation policy. Frontiers in Ecology and the Environment 8: 84–91.

[MA] Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute.

Mace GM, Norris K, Fitter AH. 2012. Biodiversity and ecosystem services: A multilayered relationship. Trends in Ecology and Evolution 27: 19–26.

McFadden D. 1974. Conditional logit analysis of qualitative choice behavior. Pages 105–142 in Zarembka P, ed. Frontiers in Econometrics. Academic Press.

Meyerhoff J, Liebe U. 2008. Do protest responses to a contingent valuation question and a choice experiment differ? Environmental and Resource Economics 39: 433–446.

Morse-Jones S, Bateman IJ, Kontoleon A, Ferrini S, Burgess ND, Turner RK. 2012. Stated preferences for tropical wildlife conservation amongst distant beneficiaries: Charisma, endemism, scope and substitution effects. Ecological Economics 78: 9–18.

Ready R, Navrud S. 2006. International benefit transfer: Methods and validity tests. Ecological Economics 60: 429–434.

Sachs JD, et al. 2009. Biodiversity conservation and the Millennium Development Goals. Science 325: 1502–1503.

Sand-Jensen K. 2007. Naturen i Danmark: Det Åbne Land [Danish nature: The Open Landscape]. Gyldendal.

Scarpa R, Rose JM. 2008. Design efficiency for non-market valuation with choice modelling: How to measure it, what to report and why. Australian Journal of Agricultural and Resource Economics 52: 253–282.

Scarpa R, Ferrini S, Willis K. 2005. Performance of error component models for status-quo effects in choice experiments. Pages 247–273 in Scarpa R, Alberini A, eds. Applications of Simulation Methods in Environmental and Resource Economics, vol. 6. Springer.

Semmens DJ, Diffendorfer JE, Lopez-Hoffman L, Shapiro CD. 2011. Accounting for the ecosystem services of migratory species: Quantifying migration support and spatial subsidies. Ecological Economics 70: 2236–2242.

[UKNEA] UK National Ecosystem Assessment. 2011. The UK National Ecosystem Assessment: Technical Report. United Nations Environment Programme World Conservation Monitoring Centre.

Veen P, Jefferson R, de Smidt J, van der Straaten J, eds. 2009. Grasslands in Europe of High Nature Value. KNNV.

Zak PJ, Knack S. 2001. Trust and growth. Economic Journal 111: 295–321.

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